2.0 Literature review

2.1 Causes and consequences of excess nitrogen in the environment

2.1.1 Causes

The uncontrolled release of nitrogen to the environment causes serious problems. Nitrate pollution in surface waters and groundwater has been attributed to wastewater effluent and agricultural runoff. For example municipal sewage, landfill leachate and wastewater from abattoirs and livestock farms are all typically high in nitrogenous compounds. Ammonia and ammonium ions occur in wastewater through the decomposition of organic matter.

Increased urbanisation and industrialisation can lead to practices, which distort the natural balances of the nitrogen cycle. This can also cause the accumulation of nitrogen compounds such as nitrate and ammonia in natural waterways.

2.1.2 Consequences

When assessing the consequences of excess nitrogen in the environment, a number of factors have to be taken into consideration. The environmental impact of wastewater discharges can be determined in a variety of ways: firstly by the composition of the effluent and its quantity, secondly by the characteristics of the receiving water body and its relative dilution, and finally by the effects to the local and more general surrounding populations (Kristensen & Krogsgaard, 1997).

Eutrophication is one of the main consequences concerning the addition of nitrogen to surface water (Middlebrooks *et al.*, 1999). Eutrophication is the enrichment of biological systems by nutrients particularly phosphorus and nitrogen. Nitrogen and phosphorus encourage the enhanced growth of algae and other plant biomasses (Reynolds, 1992), which subsequently contribute to a reduction in dissolved oxygen in the water. Ammonia is also an oxygen-consuming compound which depletes dissolved oxygen (DO). This depletion of DO in aquatic systems causes severe problems for higher forms of aquatic life.

Accelerated eutrophication and algal blooms in water bodies have caused concern over the deterioration in water quality for both recreation; seen to cause skin irritations and gastro-intestinal problems, and drinking; causing unpleasant tastes and odours. It also affects the riverine ecology, particularly in the east and the south of England. Climatic variability in the UK in the last 50 years has shown that there are greater climatic extremes causing low summer flows and high winter flows (Marsh & Sanderson, 1997). These low summer flows reduce the river's capacities for diluting wastewater input resulting in elevated nutrient concentrations which may promote the development of algal blooms (Neal *et al*., 2000). Such blooms have been observed in UK rivers including the Yorkshire Don (Pinder *et al*., 1997). Increasing human populations found in these areas also compound these pressures.

Cotman *et al.* (2001) found that effluents from municipal wastewater treatment plants contained high levels of organic nitrogen and ammonia. These high concentrations caused excessive growth of algae and macrophytes during warmer temperatures and the river became supersaturated with oxygen, particularly during the day when algal photosynthesis is high.

In addition to these effects of eutrophication, there are other consequences of high concentrations of nitrogenous compounds in the aquatic environment. Barnes & Bliss (1983) found that free ammonia (NH_3) is toxic to aquatic organisms, especially higher organisms such as fish, due to bioaccumulation. Concentrations as low as 0.5mg/l can be lethal. Middlebrooks *et al.* (1999) also found that ammonia nitrogen in low concentrations adversely affects some young fish in the receiving waters, while Wong *et al.* (2003) concurred that ammonia, nitrite and nitrate are toxic to aquatic life. Hickey *et al.* (1989) found this was particularly true where the receiving waters had dense aquatic plant growth and daytime photosynthesis caused temporarily high pH. Hickey & Vickers (1994) found that ammonia toxicity was of particular concern the native invertebrates in New Zealand. Excess ammonia in the environment also has an affect on acid rain, having subsequent effects on crops and farm workers (Zimmo *et al*., 2003).

The nitrate form is not toxic to living organisms. However after conversion nitrate becomes nitrite, which is toxic, particularly to infants. In the body, nitrite can oxidize iron (II) to form methemoglobin. Methemoglobin binds oxygen less effectively than haemoglobin. This lack of oxygen in infants has a number of repercussions such as shortness of breath, diarrhoea, vomiting and in extreme cases even death (Kelter *et al.*, 1990), also known as infantile cyanosis or blue baby syndrome.

Therefore nitrogen removal in wastewater treatment processes is important, especially in low-cost natural systems such as wastewater stabilisation ponds which are widely used in developing countries for both large and small populations and in industrialized countries mainly for small populations.

2.2 Wastewater Stabilisation Ponds (WSPs)

Wastewater stabilisation ponds are a set of shallow inter-connecting basins with earthen embankments. They are a form of sewage treatment which provides either be partial treatment or full treatment. The WSP system can either be a single series of anaerobic, facultative and maturation basins or several in parallel. The size and dimensions of the basins can vary depending on their roles. Anaerobic ponds are commonly 2-5m deep, whereas the facultative and maturation ponds are shallower, being 1-2m deep and 1-1.5m deep respectively (Mara & Pearson, 1998).

The retention time in waste stabilisation ponds is significantly longer than in traditional treatment. Wastewater continually flows into the ponds under gravity, and is retained for several days, as opposed to several hours; from which a well treated effluent is released (Mara & Pearson, 1998).

Boller (1997) described waste stabilisation ponds as a low-rate form of waste treatment as they have low loading rates, oxygenation rates and biomass concentrations compared with more advanced forms of sewage treatment. The effluent from ponds is found to have low microbial content, low biochemical oxygen demand (BOD) and low suspended solids (SS) concentrations. Often these levels are found to be on a par with, if not better than conventional sewage treatments when operated under suitable environmental conditions. They can be used as the sole wastewater treatment facility, or in combination with other treatment processes, for example as a polishing pond to improve effluent quality. Their longer hydraulic retention times enable them to cope well with fluctuating loads (Mara *et al.*, 1992).

2.2.1 The different types of WSPs

There are a number of different types of wastewater stabilisation ponds. Each type has its own functions and processes. The biological, physical and chemical processes that occur in the ponds are specific to each whether they are anaerobic ponds, facultative ponds or maturation ponds.

2.2.1.1 Functions

In general anaerobic and facultative ponds are aimed at BOD removal, whereas maturation ponds are aimed more at pathogen and nutrient removal.

BOD removal ranges between 50% and 90% depending on a number of factors including design of the basins and local climate (Mara & Pearson, 1998).

Anaerobic ponds are very efficient at removing BOD from wastewater, and their use can reduce the amount of land required for the WSP system. BOD removal is their primary function and at low temperature they can achieve BOD removal up to 40% (Mara & Pearson, 1998). The main disadvantage of anaerobic ponds is the risk of odours. Hydrogen sulphide is the main concern in this case, which is produced as a product of the anaerobic reduction of sulphate. However with proper design and loading rates odour should not be a problem (Mara & Pearson, 1998).

There are two forms of facultative ponds: those that receive raw wastewater (primary) and those that receive settled wastewater (secondary). Facultative ponds, like anaerobic ponds, are designed to remove BOD; however, unlike anaerobic ponds, they permit the growth of an algal population. The photosynthesis of these algae supplies oxygen for the pond bacteria, which is needed to remove the BOD. The high pH values that are usually found in these ponds are lethal to faecal bacteria and are also found to be effective in the removal of nitrogen (Mara & Pearson, 1998).

Maturation ponds receive their wastewater from the effluent of facultative ponds. The size of maturation ponds depends on the quality of the final effluent required. Maturation ponds are seen as a polishing stage bringing down the bacterial quality to a desired level. BOD removal in maturation ponds is generally low (Mara & Pearson, 1998). They are designed to be more efficient at the removal of pathogens and nutrients such as nitrogen.

2.2.2 Wastewater stabilisation ponds in the United Kingdom (UK)

The main disadvantage of wastewater stabilisation ponds is the large land area they require. In developing countries, this disadvantage is less significant compared to the advantages it bears. However, in Britain, where the population density is often higher than these countries, land use has to be taken into consideration. The number of investigations into pond systems in the UK is low. This is especially evident when compared to other European countries and America where these systems are used more extensively.

There is a misconception that WSPs only perform well in tropical climates, and although they do perform better at higher temperatures, that does not mean that they should be disregarded in colder climates such as in the UK. There are a number of countries with temperatures more severe than the UK that operates WSPs effectively, including Canada, Alaska and Scandinavia (Bond, 1998). The use of WSPs in the UK has so far been limited to privately owned systems or as a polishing pond in wastewater treatment plants operated by the UK water companies and authorities. All these pond systems produce an effluent quality within their consent conditions (Mara *et al*., 1996).

In The United States about 7000 WSP systems are used for treatment of municipal wastewater, irrespective of climate which ranges from subtropical to arctic (Middlebrooks *et al*., 1999). France has more than 2500 systems in operation; the highest in Mediterranean Europe, and Germany has more than 1000 (Darlow, 2000). Israel uses WSPs as its first choice for wastewater treatment, particularly given the need for the reuse of wastewater in irrigation (Mara & Pearson, 1998). But unlike much of Europe and parts of North Africa, the water companies and authorities in the UK do not use WSPs for full wastewater treatment.

Temperature is an important factor taken into consideration when assessing the performance of WSPs. Freezing temperatures are not detrimental to the performance of the ponds as the bacteria in the ponds are able to survive extremely cold temperatures. Gloyna (1971) found that ice forming on the surface could benefit the performance as it acted as insulation to the pond environment.

2.2.3 The role of algae

Algae are essential in natural aquatic ecosystems. There, they transfer solar energy into biomass, produce oxygen which can in turn be used by other aquatic organisms and are needed in the cycling of chemical elements. Their functioning depends on a variety of factors such as sunlight intensity, temperature and nutrients in the water (Walsh & Merrill, 1984).

Algae are excellent indicators of pollution. Most exist as complex colonies of varying genera and species, of which there are variations in their tolerance to pollution. Palmer (1969) found that green algae and blue green algae (strictly, cyanobacteria) were the most tolerant of pollution.

The presence of algae affects the environmental conditions within facultative and maturation ponds. The diurnal variations in light intensity and temperature affect algal photosynthesis. During the day the higher temperatures and light intensity induce higher photosynthetic activity, increasing the levels of dissolved oxygen. At the highest point of algal activity, rapid photosynthesis uses more carbon dioxide which affects the CO_2/carbonate/bicarbonate equilibrium. Carbonate and bicarbonate ions dissociate as a result and hydroxyl ions accumulate, increasing the pond pH (Mara & Pearson, 1998).

$$2HCO^-_3 \longrightarrow CO^{2-}_3 + H_2O + CO_2$$
$$CO^{2-}_3 + H_2O\ 2 \longrightarrow 2OH^- + CO_2$$

A higher pH can have a number of advantages including killing faecal bacteria and increasing nitrogen removal. During the night period, photosynthesis ceases and algal respiration begins consuming some of the oxygen present in the water (Mara & Pearson, 1998). This lack of photosynthesis causes the pH to drop during the night.

The algae that tend to dominate facultative ponds are the motile algae, such as *Chlamydomonas, Pyrobotrys* and *Euglena*. This is because they can choose the optimum vertical level in the pond that they require, depending on light intensity and temperature (Mara & Pearson, 1998). In maturation ponds the algal population is more diverse and non-motile forms become more common. A higher quality effluent in the ponds results in more diverse algal populations. Therefore less stratification is seen in the maturation ponds.

The wind velocity is seen to have a significant effect on the interactions in the pond including that of the algae. A higher wind speed will increase the vertical mixing of the pond water. A good amount of mixing will ensure uniform distribution of oxygen, algae and bacteria, producing a better final effluent. If there is insufficient mixing then the algae can settle in a thinner band causing large fluctuations in effluent quality if the algal band is moving above and below the effluent take out point (Mara & Pearson, 1998).

2.3 Nitrogen removal

Currently, cost is a major factor when considering treatment options for nitrogen and phosphorus removal from wastewater, for example in the activated sludge process (Gijzen, 2001). Treatments like this would be too costly for developing countries, so alternative effective mechanisms for nutrient removal from wastewater would be financially more appropriate. While removal of phosphorus can be achieved chemically and biologically, nitrogen removal is done almost exclusively biological (Pochana & Keller, 1999).

Wastewater stabilisation ponds are renowned for effective BOD and suspended solid removal. However the performance of WSP for nitrogen removal is less reliable. This has been seen particularly in temperate regions, where nitrogen removal can be effective in the summer months but greatly reduced in winter (Hurse & Connor, 1999). Generally ammonia concentrations in anaerobic ponds are greater than in the raw wastewater due to the hydrolysation of organic nitrogen.

This unreliable performance of nutrient removal has triggered stricter legislation to be introduced concerning the levels of nutrients in the wastewater effluent. The European Urban Waste Water Treatment Directive (CEC, 1991) requires that effluents from populations over 10,000 people should not contain total nitrogen levels higher than 10mg/l by 2005. It is estimated that less than 30% of treated wastewater in the EU was getting tertiary treatment in 1998 (European Environment Agency, 1998).

2.3.1 The Nitrogen Cycle

Nitrogen can exist in different forms depending on its oxidation states. The principal forms of nitrogen are organic nitrogen, ammonia (NH_4^+ and NH_3), nitrite (NO_2^-), and nitrate (NO_3^-). Although nitrogen gas makes up 79% of the atmosphere, it is unavailable to most living organisms in this form. Only a select group of micro-organisms, such as *Rhizobium*, *Azotobacter* and *Clostridium*, can fix nitrogen gas into a biologically usable form. Ammonia is the product of this nitrogen fixation which is essential for the formation of plant proteins.

Nitrate and nitrite are nitrogenous compounds of a higher oxidation state. Through the process of nitrification, another group of micro organisms, such as *Nitrosomonas*, *Nitrosospira* and *Nitrosovibrio*, are capable of oxidizing the free ammonia produced during nitrogen fixation to nitrite, and then *Nitrobacter*, *Nitrospira* and *Nitrococcus* go on to oxidize the nitrite to form nitrate. During these processes the terminal electron acceptor for the respiratory chain of both ammonia and nitrite is oxygen. This demonstrates the necessity of high oxygen levels for these processes to function effectively.

The nitrate can then be reduced by micro-organisms such as *Pseudomonas*, *Alcaligenes* and *Paracoccus*, during denitrification. This reduces to nitrogen gas and other nitrogen compounds (Wong *et al*., 2003).

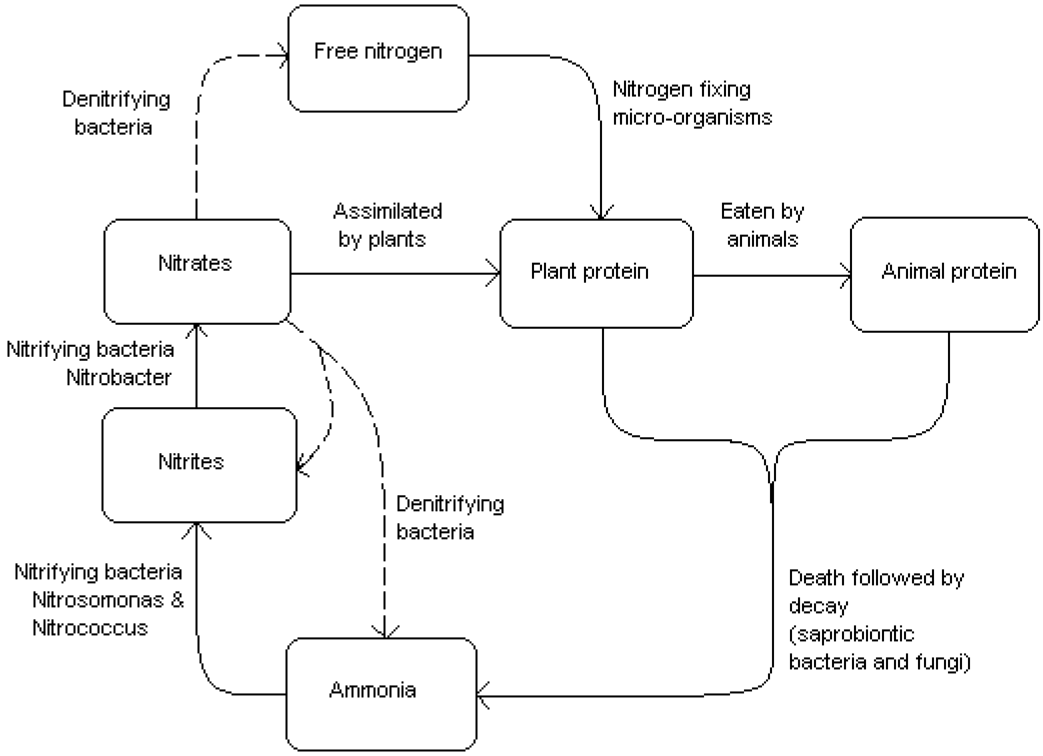

Figure 2.1. Diagram of the nitrogen cycle

2.3.2 Mechanisms of nitrogen removal in WSP

There is an on-going debate over the mechanisms by which nitrogen is removed from waste stabilisation ponds. The research conducted in this area has yielded results that are often conflicting and it is unclear which process is responsible for the majority of nitrogen loss. There are three common processes of nitrogen removal which have been assessed in previous investigations. These include ammonia volatilisation, ammonia assimilation into algae with subsequent sedimentation, and nitrification coupled with denitrification.

Nitrogen removal requires a number of key aspects to perform most efficiently. These include significant dissolved oxygen concentration, temperature sensitivity and possible nitrate build up at high temperatures (Bryant *et al.*, 1997). Zimmo *et al.* (2004a) found that lower temperatures affected the denitrification rates due to the reduction of bacterial activities.

High pH sensitivity was also found to be significant by Bryant *et al.* (1997), who found the best removal rates at a pH of 7.3. At values above or below this level ammonia removal was reduced; however, an increase in pH did not adversely affect the amount of ammonia removed as much as a decrease in the pH did. Wild *et al.* (1971) observed that the increased photosynthetic activity of phytoplankton elevated the pH by consuming the acidic carbon dioxide in the pond water.

The diurnal patterns and seasonal changes have an influence on all these factors. McLean *et al.* (2000) found that dissolved oxygen concentrations were at the lowest point from 6-8am and peaked at about 4pm, while Mikkelsen & Datta (1979) found that pH varies between night and day, with the values found to be considerably lower at night. Lai & Lam (1997) observed that during the hotter seasons nitrogen removal was more efficient compared to the colder seasons of autumn and winter.

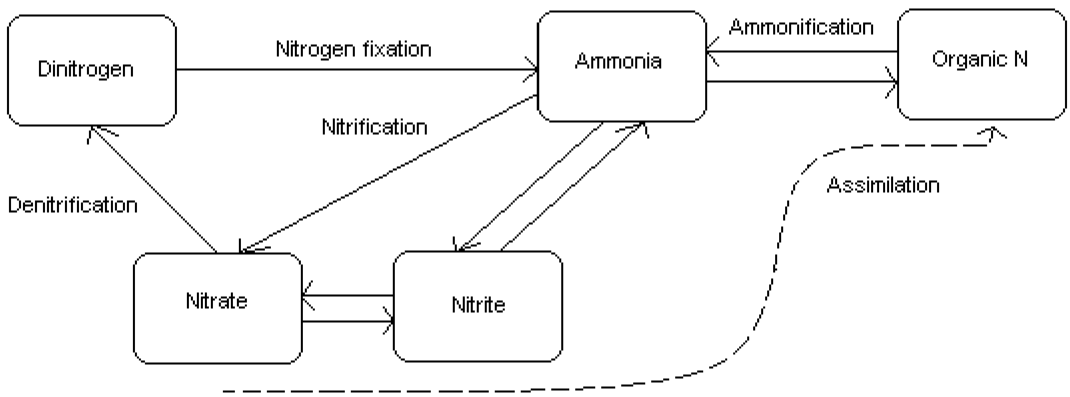

Figure 2.2. Application of the nitrogen cycle in wastewater treatment (Wong *et al.*, 2003).

2.3.2.1 *Nitrification and denitrification*

Nitrification is also known as the biological oxidation of ammonia to nitrate. It is for this reason that scientists have measured nitrate concentrations in WSP, as an indication of nitrification (Craggs *et al.*, 2000; Bryant *et al.*, 1997).

However many research workers have investigated the possibility of the nitrate undergoing further conversion through denitrification. Denitrifying bacteria tend to thrive in oxygen-deprived conditions; they use the oxygen from nitrate as the final acceptor in the hydrogen carrier system. In the presence of denitrifiers, nitrate is broken down to produce nitrogen gas (N_2) and alkalinity (HCO_3). This may therefore explain why nitrate may not be detected in the pond water. Due to the low levels of oxygen in the pond water, after nitrification the denitrification process could begin, causing a nitrification and denitrification cyclical pattern. Other factors also contribute to this such as algal respiration and bacterial degradation of organic matter.

Although many investigators argued that low nitrate concentrations demonstrated the lack of nitrification in wastewater ponds, Santos & Oliveira (1987) and Zimmo *et al.* (2004b) demonstrated that denitrification is one of the major mechanisms of nitrogen removal. These authors consider that it is through denitrification that nitrate is converted into oxygen, nitrogen gas and alkalinity, explaining why nitrate could not be detected in some of the previous investigations.

Zimmo *et al*. (2003) found that ammonia volatilisation accounted for only 2-3% of nitrogen removal from their ponds and concluded that nitrification/denitrification accounted for the majority of nitrogen removal. Additional work by Zimmo *et al.* (2004a) also found that in the algal-based ponds they investigated, ammonia volatilisation was seen to be the least significant mechanism of nitrogen removal only accounting for 1.1% of its overall removal. They found that denitrification and sedimentation were the major removal mechanisms.

Lai & Lam (1997) made some important observations in their ponds in Werribee, Melbourne. They found a correlation between the retention times of their ponds and the amount of nitrate observed. The study system consisted of eight ponds connected in a series. Raw unsedimented sewage passed into pond 1, and then enters into the successive ponds through overflow outlets. Lai and Lam (1997) found that there was a general increase in dissolved oxygen and chlorophyll a content from pond 1 to 8, and the treated sewage also became more alkaline. During the autumn/winter period, they found that in ponds 4-8, nitrate was being produced while ammoniacal nitrogen was decreasing relatively. The wastewater in the later ponds had been in the system for a longer period than the earlier ponds, so the equivalent retention time is a lot longer than most waste stabilisation ponds in operation in tropical countries; with a retention time less than 40-50days. From this they concluded that nitrification was taking place; however, because there was an accumulation of nitrate, they also assumed that denitrification was not occurring. They deduced from this that the longer retention time enabled the algae to better establish themselves; most of the BOD has been biodegraded so the nitrifiers are not competing for oxygen with the heterotrophic bacterial populations; therefore as they thrive, the intensity of photosynthesis can be so high that aerobic conditions can spread all the way down to the interface with the anoxic layer. This meant that the anoxic conditions usually present tend not to occur as they are shallower and well mixed, possibly preventing denitrification from occurring. While this investigation is important as they were the first to observe nitrate levels in ponds, they also found unusual results in their seasonal variations and their pH significance. They found more nitrogen removal during the autumn and winter months than the summer months, and concluded pH was of no significance to nitrogen removal. However, it is important to note that they only took 12 pH readings over the year and did not take into account diurnal variations or consider the effect of the algae not photosynthesizing but respiring during the night.

Hussainy (1990) also showed that diurnal patterns in dissolved oxygen can occur from the phytoplankton blooms. Denitrification requires anoxic conditions, and he found that, during the night when photosynthesis was reduced, the DO level could fall to levels low enough to permit denitrification to occur. This diurnal pattern resembles the aerobic and anaerobic conditions used in the activated sludge reactor for nitrogen removal.

However, a number of investigators have concluded that nitrification is not a major method of nitrogen removal from wastewater, no evidence of nitrate was found in the pond systems (Mara & Pearson, 1986; Reed, 1985). Mara & Pearson (1998) found that populations of nitrifying bacteria are very low in waste stabilisation ponds because of the lack of physical attachment sites in the aerobic zone which suggested that nitrification does not occur readily. Stone *et al*. (1975) also found that a major factor limiting nitrification was the lack of permanent aerobic attachment surfaces for nitrifying bacteria, particularly in facultative ponds. They also found that there was a lag period between sufficient dissolved oxygen levels being established and nitrification proceeding so that nitrification can only develop if long-term conditions prevail. McLean *et al*. (2000) found that active nitrification was confined to the upper limits of the ponds due to the greater light intensity creating an aerobic zone; however, sometimes nitrifying bacteria populations spread across the water column and thrived, but these periods were short lived and unpredictable and usually only occurred in hotter months. This evidence against the occurrence of nitrification implies that some other method of removal is responsible for the majority nitrogen removal from the wastewater.

2.3.2.2 Ammonia volatilisation

Like the other mechanisms of nitrogen removal, ammonia volatilisation is known to be dependant on a number of key factors. The differences in wastewater pH and temperature largely determine the magnitude of ammonia volatilisation losses from the WSP surface. Pano & Middlebrooks (1982) proposed that the carbon dioxide consumed by algae during photosynthesis exceeds the amount supplied by organic degradation, which increases the pH. Raised pH levels shifts the equilibrium from ammonium ions towards free ammonia able for volatilisation and vice versa.

$$NH_3 + H_2O \rightleftharpoons NH_4^+ + OH^-$$

$$NH_4^+ \rightleftharpoons NH_3 + H^+$$

Middlebrooks *et al*. (1999) found that at a pH of 8, 95% of nitrogen is in the form of ammonium and at a pH of less than 8, ammonia volatilisation is less effective. The rate of volatilisation also increased at higher concentrations of ammonia-N and TKN.

Wind velocity over the top of the pond surface is seen to be a key factor as it enhances the mass transfer coefficient. The greater the wind velocity, the more ammonia volatilisation is thought to occur due to increased vertical mixing. Maynard *et al*. (1999) reported that for volatilisation to take place a pH in excess of 10 was needed to work effectively; however, they also reported that with increased wind velocity and vertical mixing, a pH of 7-8 could create effective volatilisation.

Toms *et al*. (1975), Pano & Middlebrooks (1982), Reed *et al*. (1995), Silva *et al.* (1995) and Soares *et al.* (1996) all proposed that ammonia volatilisation was the major mechanism for nitrogen removal from ponds. Oron *et al.* (1987) could not account for 50% of the nitrogen removal in their ponds, and they too concluded volatilisation was the major contributing mechanism.

Van der Steen *et al*. (1998) concluded that 73% of nitrogen removal in the pond was due to either ammonia volatilisation or denitrification. As there were almost always aerobic conditions in the pond, they concluded that denitrification could have only been mildly significant so, volatilisation appeared to be the major mechanism of removal. They found that nitrification/denitrification accounted for as little as 3% of total nitrogen removal.

Although Zimmo *et al*. (2003) found that ammonia volatilisation rates fluctuated with pH and temperature levels as previously thought, they found that ammonia volatilisation only accounted for a small amount of the total nitrogen removal. In their algal based ponds (ABPs) and duckweed based ponds (DBPs), volatillsation accounted for less than 2-3% of total nitrogen removal.

2.3.2.3 Algal uptake and sedimentation

Nitrate is the main form of nitrogen that is directly available to plants. The ammonium compound (NH_4^+) can also be absorbed by many plants. However, nitrites are toxic to most plants and nitrogen gas cannot be used by them.

Nitrogen compounds are taken up by the algal cells in the ponds, and after the algae die they settle to the bottom of the pond. Some nitrogen remains fixed in the algal matter because approximately 20% of the algae are non-biodegradable so the nitrogen associated with this amount becomes immobilised in the sediment of the pond. Any fraction of nitrogen that is in the biodegradable part of the algal mass diffuses back into the pond water (Mara & Pearson, 1998).

Many concluded that the role of algal uptake and sedimentation was comparatively insignificant compared to volatilisation and nitrification. The majority of investigations saw only a few percent of total nitrogen removal in dried algal samples.

However there are some investigations which argued that algal sedimentation caused high rates of nitrogen removal. Ferrara & Avci (1982) found that sedimentation was the main removal pathway for nitrogen. They found that the abundant phytoplankton had a preference for nitrogen in the form of ammonium. This was confirmed by Zimmo *et al*. (2004a) who found that the largest nitrogen flux was due to the sedimentation of particulate organic nitrogen through the decaying algal biomass. At higher temperatures when algal growth was thriving, there was 15% more nitrogen accumulated in the sediment. Strang (2001) suggests that biomass assimilation would be a major mechanism of removal in facultative ponds which are heavily loaded, but less significant for maturation ponds.

Lai & Lam (1997) found in some of their ponds that inorganic nitrogen was converted to organic nitrogen in algal cells through photoplankton uptake and growth; however, they concluded that algal uptake could not account for the complete removal of nitrogen.

From these examples of investigations it is clear that there is much conflicting evidence regarding which mechanisms are responsible for the majority of nitrogen removal. The two main mechanisms which have been considered are nitrification/denitrification and volatilisation. Although there has also been proof that algal uptake and sedimentation occurs, few investigations have considered this mechanism to be the primary form of removal. That would be an acceptable assumption considering most investigations found a combination of all of these mechanisms were present in varying percentages (Van der Steen *et al*., 1998).

2.3.3 Ammonia Stripping Model

Pano and Middlebrooks (1982) developed a mathematical model designed to assess the theoretical amount of ammonia nitrogen removal from a pond, taking into account three variable factors; pH, temperature and hydraulic loading rate. This ammonia stripping model made the assumption that nitrification was not a significant part of nitrogen removal, although this assumption is not necessarily correct, the model can still calculate with accuracy, ammonia removal from a pond.

The ammonia stripping model was calculated using this equation:

$$C_e = \frac{C_i}{\{1 + [(A/Q)(0.0038 + 0.000134^T) \times \exp((1.041 + 0.044^T)(pH - 6.6))]\}}$$

where:

C_e = NH_3-N concentration for effluent (mg/l)

C_i = NH_3-N concentration for influent (mg/l)

A = Pond surface area (m^2)

Q = Flow (m^3/day)

T = Temperature (°C)

The results of the ammonia stripping model can be compared to the actual results obtained during an investigation. This would provide more conclusive evidence on the accuracy and relevance of the results obtained.